Review of science used by Massachusetts Department of Environmental Protection to develop statewide streamflow policy.

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1 Introduction

The Massachusetts Department of Environmental Protection (MassDEP), in conjunction with other government agencies and stakeholders, is developing a Sustainable Water Management Initiative (SWMI) that defines Safe Yield withdrawal estimates and stream flow criteria (SWMI 2012). A major component to the draft SWMI (2012) framework is application of the United States Geological Survey's publication "Factors influencing riverine fish assemblages in Massachusetts" (Armstrong et al. 2011). Because of the framework reliance on this scientific work, MassDEP has requested a technical review of both: a) Armstrong et al.'s (2011) report; and, b) a rebuttal document to the report (TRC 2012). The technical review would assist MassDEP in determining appropriateness of the current draft SWMI.

The following report provides a technical review of Armstrong et al.'s (2011) report and the rebuttal document (TRC 2012). The review does not assess whether the draft SWMI (2012) is supported by the Massachusetts Water Management Act, other State legislation or Federal law.

As requested by MassDEP, objectives of the report are:

1. Provide a technical review of the USGS report by Armstrong et al. (2011). Are methods and results scientifically sound and robust? Are there concerns that must be considered when applying the science?

2. Has the science in Armstrong et al. (2011) been applied in a technically appropriate manner to establish water management policies within the draft SWMI (2012)?

3. Provide a technical assessment of the TRC rebuttal document (TRC 2012) that raises concerns related to both the Armstrong et al. (2012) report and how this information has been applied to establish water policy.

The report is structured into sections that follow the three objectives.

2 Review of USGS report "Factors influencing riverine fish assemblages in Massachusetts" (Armstrong et al. 2011)

2.1 Are methods scientifically sound and robust?

Quantile regression and general linear models

The use of quantile regression is an appropriate approach to relate explanatory variables (e.g., IC – impervious cover or AUGgwWp – August groundwater withdrawals as a percentage of median unaltered flow) to a response variable (e.g., RFFA – relative fluvial fish abundance) when it is expected there are many alternate and unmeasured variables that also affect the response variable (Cade and Noon 2003). If impervious cover or withdrawals are one of several limiting factors, then quantile regression is an appropriate tool to assess the rate of change in fluvial fish abundance with these explanatory variables. Lower quantiles (e.g., median) or mean responses (as used in general linear models [GLM]) will tend to show weaker or even no response if multiple unmeasured variables influence the response variable.

The difference between quantile and GLM regressions is evident when comparing Figure 13a to Figure 20a. The 90th quantile regression line of Figure 13a has a steeper slope than the GLM regression in Figure 20a. The implication being biological effects predicted from a given flow alteration using the GLM model may be underestimated. Or in other words, if quantile regression were used, the same alteration in August median flow would be predicted to have a larger biological effect than the GLM model.

Relative fluvial fish abundance as a response variable

Assignment of fish to the fluvial category is justified both on biological life-history characteristics (Table 1) and observed statistical distribution of fishes among the 669 sampling sites. Agreement between cluster analysis and nonmetric multidimensional scaling (NMDS) provides good support for using Relative Fluvial Fish Abundance as a response variable (Figure 11).

I do have questions regarding the removal of non-resident and anadromous fish species from the analysis (page 6). I agree with Armstrong et al.'s (2011) argument for removal as their presence can be temporary and not reflective of local habitat conditions. However, non-resident or anadromous species may displace resident fishes (including fluvial specialists) when they are present through: a) competition for similar niche space; or, b) predation. A simple way to determine support for this hypothesis would be: a) plot relative fluvial fish abundance against relative anadromous species abundance; and, b) use quantile regression to determine whether a statistical relationship exists between the two variables. If there is no relationship, then the hypothesis is not supported; if there is a relationship, further work or explanation would be required to assess potential for a casual relationship.

Multicollinearity among explanatory variables

Armstrong et al. (2011) are correct in stating inclusion of multiple correlated explanatory variables can lead to incorrect signs and magnitudes of regression coefficients as redundant variables may spuriously capture residual variance. Therefore, highly correlated explanatory variables should be either: a) reduced into a single composite metric (e.g., principal component factors); or, b) use only one of the explanatory variables in the statistical analysis. Regardless of choice (either a or b), one is still left with the task of explaining which of the explanatory variables provide a mechanism that causes the response variable to change. To help determine mechanistic support for different explanatory variables, plausible hypotheses should be listed along with their biological support.

For the work of Armstrong et al. (2011), the correlation between east/west location (Outlet X) and impervious cover (IC) or August groundwater withdrawals (AUGgwWp) is a potential concern regarding collinearity amongst the three explanatory variables. I agree with the author's decision to reduce highly correlated explanatory variables to a single variable by eliminating Outlet X (page 27). The concern is the removed variable (Outlet X) may contain the actual mechanistic relationship driving the response. This is clearly acknowledged by the authors on page 49, their recommendation to include spatial structure (essentially east/west location) into the analysis would help evaluate this concern.

A rather simplistic way to analyse spatial structuring of the data is to split the state of Massachusetts in half around 72°W and repeat the quantile regression and GLM analyses on sites in each half. If relationships hold in each half, then there would be less concern spatial location is related to fluvial fish abundance. A problem in this approach is that reduction in sample sites (by splitting the data) will reduce statistical power.

2.2 Are results scientifically sound and robust?

The results of Armstrong et al. (2011) show a strong negative relationship between impervious cover (IC) or August groundwater withdrawals (AUGgwWp) and relative fluvial fish abundance (Table 8a; Figure 20a). There is a small positive bias to the model but it is consistent across sites (Table 8b). However, the model has only limited predictive ability when determining relative fluvial fish abundance at a particular site. Only about 18% of variability in observed fluvial fish abundance can be explained by: channel slope (CHSLP); August groundwater withdrawals (AUGgwWp); percent area of buffer as wetland (pB(Weg_al); and, impervious cover (IC). Given >80% variability in fluvial fish abundance at a site is not explained by the model, one should not be surprised that large discrepancies between site-specific model predictions and observed data. As stated by Armstrong et al. (2011), the important result from the analyses is the significant trend in fluvial fish abundance across the gradient of each explanatory variable (page 49).

There were two rather minor errors in the report:

1) I do not agree with the conclusion that "*few fluvial fish remain at high rates of withdrawal (approaching 100%)*" as stated in the abstract and again on page 50. The statement oversells the predictive ability of the GLM at determining fluvial fish abundance at a particular site. Recalling >80% unexplained variance in site-by-site predictions, I believe the more correct statement is "*sites with high rates of withdrawal (approaching 100%) tend to have significantly fewer fluvial fish than sites with lower withdrawals.*" Some sites with high withdrawals may have high abundance of fluvial fish but they are much less common, while sites with low withdrawals can still have few, or no, fluvial fish (see Figure 13a).

2) The statement "*a one-unit (1 percent) increase in the percent depletion of August median flow would result in a 0.9-percent decrease in the relative abundance (in counts per hour) of fluvial fish.*" is not entirely correct. Given the GLM is a Poisson model, the log-linear relationship is of the form $y=e^{bx+a}$, where y is the response variable, x the explanatory variable, b the slope parameter and a the intercept. For the fluvial fish abundance model shown in Table 8, only at $x=0$ does the above statement hold true. As x increases, the rate of change in fluvial fish abundance declines (Figure 1).

Overall, I found the results of Armstrong et al. (2011) to be sound and robust given material presented and inherent to limitations of observation-based studies, as discussed in Section 2.1.

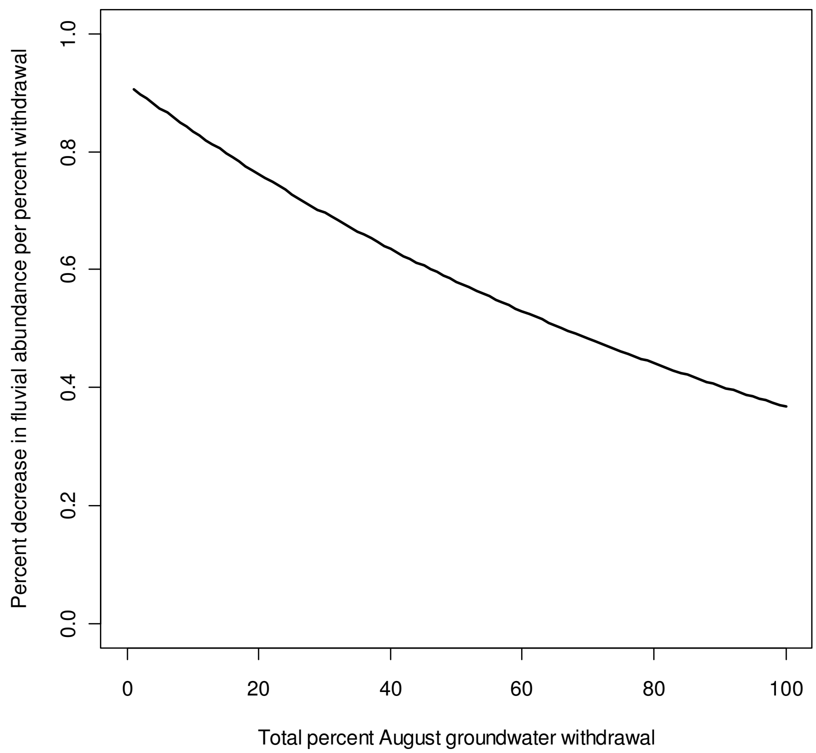

Figure 1 – Percent change in fluvial fish abundance for each percent increase in August groundwater withdrawal as a function of the total August groundwater withdrawal using the Poisson (log-linear) model in Table 8 of Armstrong et al. (2011).

2.3 Concerns using the science for management applications?

The dose-response model (i.e., August groundwater withdrawal versus fluvial fish abudance) of Armstrong et al. (2011) provides a useful tool for regional water management applications. The significant trend (Figure 20a) can be used to manage August groundwater withdrawals to meet an objective defining acceptable change in fluvial fish abundance. However, application of the relationship has two important sources of uncertainty:

1) Site-by-site predictive ability of the model is poor. The quantile regression plot (Figure 13a) shows August groundwater withdrawals only provide an upper limit to expected fluvial fish abundance. Other explanatory variables will influence actual abundance, including variables not measured in Armstrong et al. (2011). For example, the model should not be used to predict the outcome of restoration efforts. Although reducing groundwater withdrawals may be necessary to increase fluvial fish abundance, it

is quite possible other limiting factors may prevent an increase in abundance if flows are restored.

2) Correlative relationships do not imply a causative mechanism. Other unmeasured (or collinear) explanatory variables may be driving the observed relationship.

The implications of these sources of uncertainty are discussed in more detail in Section 3.

3 Application of USGS report (Armstrong et al. 2011) to water management policy in Massachusetts

3.1 General

As discussed in Section 2.3, there are two important sources of uncertainty when applying an observed empirical model to regional management.

The model is not a good predictor of individual sites

Perhaps the best way to discuss this is through an example. Suppose regional withdrawal limits are set at 25% to maintain changes in the fluvial fish community at an acceptable level. Because many sites have no fluvial fish even though current groundwater withdrawals are zero (see Figure 13a); it might seem pointless to impose a withdrawal limit to protect a non-existent fluvial fish community. Application of an empirical (i.e., data-driven) model at a regional scale for water management will need to discuss and deal with discrepancies between regional objectives and site specific conditions. For instance, it is reasonable to assume fluvial fishes are present downstream of sites where they are absent.

Correlative relationships do not imply causation

The work of Armstrong et al. (2011) is an empirical study exploring correlative relationships between measured explanatory and response variables. Correlative-based studies do provide important information toward understanding and managing environmental flows (Locke et al. 2008). However, when using correlative-based studies, two fundamental questions must be addressed:

1) Does a plausible mechanistic hypothesis support the observed relation?
2) Are there other plausible hypotheses (i.e., unmeasured or collinear variables) that may also explain the observed relation?

With respect to the first question, Armstrong et al. (2011) provide a thorough and well referenced discussion supporting the hypotheses that fish communities depend on flow regimes, and hence water withdrawals (see Page 5). For the second question, there remains uncertainty regarding collinearity between east/west position (Outlet X) and August groundwater withdrawal (AUGgwWp). Certainly, east/west position on its own does not provide a good mechanistic explanation for patterns in fluvial fish abundance; but, are there other mechanisms that follow the same spatial pattern? One possibility might be human density or angling effort and resulting fish mortality. However, angler mobility and information sharing tends to homogenise fish densities at a regional scale (Post et al. 2002). Furthermore, most of the fluvial fish species in Armstrong et al. (2011) are not sport fishes and large catchable sized trout (>200 mm) were excluded from the analysis. A second potential hypothesis is that anadromous fishes (that were excluded from the analysis) may be more abundant in the eastern part of Massachusetts. If these fishes occupied niches of fluvial residents or preyed on them, then abundance of the fluvial fishes could be negatively related to anadromous fish abundance. As discussed in Section 2.1, this hypothesis could be further explored by looking for a pattern in distribution between fluvial and anadromous fish abundance using the existing data.

3.2 Draft sustainable water management initiative (SWMI 2012)

The draft sustainable water management initiative of Massachusetts establishes five biological effect categories that represent percent changes in fluvial fish abundance from changes in flow or impervious cover (SWMI 2012). Taking some liberty to paraphrase, discrete changes in fluvial fish abundance from withdrawals or impervious cover are:

- categories 1 and 2 (0 – 15% decline in the range of fluvial fish abundance) represent low risk situations where fluvial fish biodiversity is maintained and communities are resilient;
- category 3 (15 – 35% decline in the range of fluvial fish abundance) is a moderate risk situation where fluvial fish biodiversity is likely maintained but sensitive species are at reduced abundance; and,
- categories 5 and 6 (35 – 100% decline in the range of fluvial fish abundance) is a high risk situation where biodiversity of fluvial fish is increasingly threatened.

While setting absolute reference values separating the categories is difficult and imprecise, the exercise is necessary to provide decision-makers with guidance on whether there interests or objectives are being addressed (Locke et al. 2008; Ohlson et al. 2010). The categories used above are logical (increasing risk with increasing change in fluvial fish abundance) and consistent with thinking used internationally to establish reference scales (IUCN 2001[1]).

Based on the defined biological risk categories, the draft sustainable water management initiative (SWMI 2012) converts the risk categories into percent August flow alterations using the fluvial fish GLM model (equation 6, Armstrong et al. 2011). Not withstanding uncertainties discussed in Section 3.1, this is an acceptable application of the available information. The biological risk categories are based on percent alteration to the scaled range of observed fluvial fish abundance over the range of August groundwater

[1] An IUCN (2001) criterion for listing a taxon as vulnerable (defined as having a high risk of extinction in the wild) is a ≥30% population decline.

withdrawals (T. Richards, pers. comm., Massachusetts Department of Fish and Game). The response variable (alteration of range of fluvial fish relative abundance) is now scaled between 0% and 100% (Figure 2). The scaling approach needs to be clearly documented within the technical appendices of SWMI (2012)[2]. Based on the biological categories, corresponding alterations to flow were calculated by SWMI (2012). As a calculation check, I was able to derive similar flow categories using equation 6 of Armstrong et al (2011) and the biological categories of SWMI (2012).

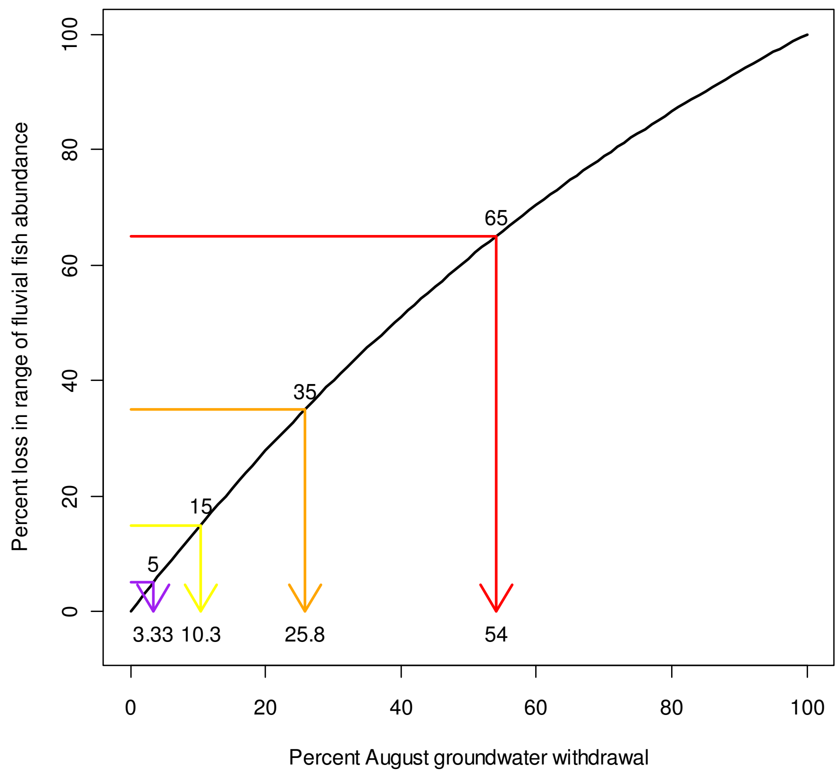

Figure 2 - Relation between percent change in relative fluvial fish abundance and percent August groundwater withdrawal using GLM equation 6 of Armstrong et al. (2011). All explanatory variables, other than withdrawals, were held constant. Coloured arrows represent biological risk categories of SWMI (2012) and the corresponding change in August flow required.

[2] Technical appendices to SWMI (2012) were not reviewed in this document.

4 Technical assessment of concerns raised in TRC (2012) rebuttal document

The final section provides an assessment of concerns raised by TRC (2012) regarding both the technical work of Armstrong et al. (2011) and its application to water policy through Massachusetts' draft Sustainable Water Management Initiative (SWMI 2012). I identified 10 concerns in TRC (2012) and these are listed, with my response, in the following sections (page numbers from the TRC [2012] document are provided in parentheses).

4.1 No quantitative reference points for species loss in biological categories (page 3)

The biological risk categories of SWMI (2012) require as much explanation on their meaning and derivation as possible. SWMI (2012) indicates the categories were based on quantile regression and input from stakeholders. Explanation of how quantile regression was used would be helpful, as well as key stakeholder input. However, the risk categories should also reflect that they are qualitative and imprecise. This is not a fault but rather a recognition of uncertainty. Quantitative values for number of fluvial species lost with increasing withdrawals is not predicted by Armstrong et al. (2011). Therefore, the risk categories can be given quantitative reference points regarding biodiversity.

4.2 Relative fluvial-fish abundance model contradicts a general principle of ecology (Pages 3 and 4)

A relationship between relative fluvial-fish abundance and river flow does not contradict any principles of ecology. If the hypothesis that fluvial fish are limited by physical area is true (i.e., more surface area of water equals more fish), we would observe a positive relationship between fluvial-fish abundance and river flow as wetted area is always an increasing (but at a declining rate) function of flow. By the same token, a decrease in flow from withdrawals would result in a decrease in fluvial-fish abundance. This is a general principle of limiting resources in either ecology or economics.

What sometimes misleads our thinking when considering flow as a limiting resource is its natural variability both within and among years. No resource in nature is constant; but, if you remove a constant amount x all the time, you shift the magnitude of its distribution by the same amount x. That is, the mean, median, range, 75^{th} percentile, etc. are x units less than without the removal.

The relative fluvial-fish abundance model of Armstrong et al. (2011) does not predict species extinctions given among year variation in summertime low flows of 100-300% for individual sites. First, the fluvial-fish abundance model does not predict species richness but rather an aggregate measure of relative abundance for all fluvial species. The species richness models (equations 4 and 5) do not have any flow alteration metrics as explanatory variables (Armstrong et al. 2011). Species richness was positively related to drainage area of a sampling site (Table 8, Armstrong et al. 2011) which implies larger streams, on average, contained more species. Second, the fluvial-fish abundance model predicts there would be (on average) a catch of 100 fluvial-fish per hour electrofishing, even with 100% alteration of August median flow from groundwater withdrawals (Figure 20, Armstrong et al. 2011).

4.3 Collinearity of Outlet X with other predictor variables (page 5, 7 and 8)

This concern has been discussed in detail in previous sections (sections 2.1 and 3.1). Briefly, plausible alternative hypotheses need to be identified and evaluated where possible. TRC (2012) identify a coupled set of hypotheses that provide an alternate explanation for fluvial fish abundance across an east/west gradient:

> 1) fluvial fish species diversity is positively related to elevation (a temperature effect?); and,
> 2) fluvial fish abundance is positively dependent on fluvial species diversity (i.e., more fluvial species means higher abundance).

Given the fluvial species richness model included impervious cover and impervious cover is correlated with elevation (Armstrong et al. 2012), hypothesis 1 can not be dismissed. Hypothesis 2 could be tested by plotting fluvial fish abundance against fluvial species diversity; or in other words, does fluvial species richness translate into greater abundance or is fluvial abundance determined by available niche space?

4.4 Selection of 40mm cut-off length (page 5)

Removing small fish (e.g., <40 mm) is normal practice for electrofishing studies as capture efficiency by electrofishing decreases rapidly for small fish. The use of relative abundance as a surrogate measure for abundance assumes constant catchability (q; proportion of population captured per unit of effort)

$C/E = qN$

where C is total catch of fluvial fish (number), E is effort (electrofishing hours) and N is fluvial fish abundance. If q is unknown but constant, then catch-per-unit effort (C/E) is directly proportional to N. Eliminating small fish from the electrofishing catch provides validity to the assumption of constant q.

However, there is potential for removal of small fish to have biased results of Armstrong et al. (2011) if sites with high August groundwater withdrawals were sampled at the beginning of the year and sites with low August groundwater withdrawals were sampled later in the year. The bias would have occurred as earlier in the year small fish would have either: a) not been captured by electrofishing because of reduced efficiency; or, been removed from the catch as they were <40mm. Whereas, fish would have been included in the abundance measure later in the year as they both grew over the 40mm cut-off and were more likely to be captured by electrofishing. The likelihood of this bias can be tested by plotting fluvial fish abundance for each site against sampling date (Julian day), if high abundance sites were sampled later in the year there is support for a sampling date bias.

4.5 Exclusion of nearby samples (page 6)

The removal of nearby sampling sites is justified. Quantile and GLM regression assume unexplained variance is independently distributed among sampling sites. Removing nearby sites helps support the assumption. For nearby sites, the method of Armstrong et al. (2011) to keep the site with the greatest drainage area (random selection in the event of ties) is not expected to have introduced a bias into their results.

4.6 Express CPUE as fish per hour (page 7)

Expressing catch-per-unit-effort (CPUE) for electrofishing as counts per hour is not as familiar to myself as counts per second or counts per minute. However, this is a simple linear scaling issue and units can easily be changed as desired.

4.7 Incorrect assignment of sub-basin to onerous permitting category based on relative fluvial fish abundance (page 8)

SWMI (2012) assigns streams to flow level criteria based on water withdrawals and not relative fluvial fish abundance; therefore, the statement as worded by TRC (2012) is incorrect. However, interesting site-by-site situations arise when applying the empirical model for regional management (see also Section 3.1). Most notably, sites with no water withdrawals and no fluvial fish would be held at <25% future withdrawals to protect fluvial fish abundance, even though no fluvial fish are present. A management framework should provide detail on why these situations arise and how they would be handled, for example:

a) although fluvial fish are absent at a site, their presence downstream is expected and withdrawal limits would be protective of these fish communities;
b) absence of fluvial fish at a site may be related to other aspects of their habitat (e.g., loss of forest cover), however this does not provide an exemption from water management requirements; or,
c) regional-based management frameworks could be adjusted with sufficient and relevant site-specific information.

Questions and concerns will always arise when applying a regional-level management framework to individual sites. The strength of a framework will depend on its ability to address the questions and concerns in a consistent and defensible manner.

4.8 Stream-size sampling bias (page 8)

As discussed in Section 4.4, the use of relative abundance (counts per electrofishing hour) assumes constant catchability (q) across sites. Armstrong et al. (2011) limited their sites to wadeable streams and excluded sites with high water or high turbidity, these are appropriate and standard measures to meet the assumption of constant catchability.

However, stream-size sampling bias could arise if larger streams have higher groundwater withdrawals and catchability decreases with stream size. The bias can be explored by plotting stream-size against August groundwater withdrawals to determine whether there is a positive correlation. This pairwise scatterplot is not shown in Figure 8 of Armstrong et al. (2011). However, drainage area (DA, a measure of stream size) and impervious cover (IC) show almost no correlation.

A second hypothesis is the observed relationship for wadeable streams can not be extrapolated to larger systems. Or in other words, the relationship between fluvial fish abundance and August groundwater withdrawals is specific to smaller systems and does not hold for the larger rivers. However, the information provided in Armstrong et al. (2011) doesn't support this hypothesis as drainage area (DA, and a measure of stream size) was not an important explanatory variable for fluvial fish abundance (Armstrong et al. 2011).

4.9 Accounting for diadromous fluvial species (page 9)

Catadromous American Eel were included by Armstrong et al. (2011) in their analysis. They were not part of the fluvial fish assemblage as MADFW has classified them as a macrohabitat generalist and this classification was supported by the multivariate analysis of Armstrong et al. (2011).

Exclusion of anadromous fish from the analysis may have eliminated an important explanatory variable for fluvial fish abundance (see Sections 2.1 and 3.1 above). This can easily be explored by plotting relative fluvial fish abundance against relative anadromous fish abundance.

4.10 RFFA versus August flow depletion on a site-specific basis (starts page 9)

Testing the relationship between relative fluvial fish abundance (RFFA) and August groundwater withdrawals developed by Armstrong et al. (2011) on a site-by-site basis is inappropriate. As stated by Armstrong et al. (2011),

> *Although the predictive ability of these models is not high, the relations characterized by the coefficients in the model are highly significant.* (page 49)

The trend of declining fluvial fish abundance with increasing August groundwater withdrawals is of importance, not the prediction of abundance at a particular site. Figure 13a (Armstrong et al. 2011) clearly shows there is substantial variability in fluvial fish abundance on a site-by-site basis. However, sites with high withdrawals are less likely to have high fluvial fish abundance (although not impossible) than lower withdrawal sites.

A simple analogy would be to measure the heights of students in a classroom and calculate the average. Although the average is a measure of central tendency and can be a useful statistic, it is entirely likely that nobody's height in the classroom equals the average; and, in fact, we could expect more than half the students to be shorter than the average if the distribution of heights is skewed to the right by a few very tall students.

5 Summary

The work of Armstrong et al. (2011) is technically competent and provides important and useful information for understanding how fish communities respond to changes in flow. As Armstrong et al. (2011) discuss, their results may be confounded by environmental and land-use variables that co-vary along an east-west gradient in Massachusetts and further spatial structuring of their analysis would be useful. I have discussed several simple explorations of the data that would help differentiate among alternate hypotheses. However, presence of confounding factors (or alternate hypotheses) will always occur for correlative studies using observational data.

Moving from science or technical-based studies to management action always incorporates uncertainty. Management actions must consider three questions:

1) What interests are to be considered?
2) What information is available to understand the impact on these interests?
3) How will uncertainty be addressed?

Uncertainty should not be used to delay or avoid natural resource problems, decisions are made by confronting uncertainty (Ludwig et al. 1993).

6 References

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